A Foundational Synthesis of Ecosystem Services Knowledge to Critically Assess and Structure Their Integration into Life Cycle Impact Assessment

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Abstract

Purpose: The integration of ecosystem services in life cycle assessment has gained increasing attention. Its operationalization remains limited and inconsistent, particularly within the impact assessment phase. This study provides the first comprehensive and structured critical review that bridges foundational ecosystem services knowledge with life cycle assessment modeling needs.

Method: We performed two complementary literature reviews: (i) a foundational synthesis of ecosystem services concepts informing (ii) a critical review of existing proposals to integrate ecosystem services into life cycle assessment. By combining and enhancing existing classifications, we provided a refined typology of integration approaches and map 43 studies onto this structure.

Results and discussion: Our analysis reveals conceptual ambiguities, inconsistent modeling purposes, insufficient consideration of ecosystem services area of protection interlinkages, and an overreliance on land-use impacts at the expense of other critical pressures. The inherent connection between natural resources and provisioning ecosystem services is overlooked by the life cycle impact assessment community.

Conclusion: This review highlights the need for a coherent, harmonized, and globally accepted framework to facilitate the integration, in life cycle impact assessment, of impacts on ecosystems' capacity to provide services. We outline key priorities for model developers, including terminology harmonization, explicit modeling purposes, improved coverage of ecosystem services categories, better representation of pressure pathways, and clarified relationships between ecosystem services and existing areas of protection. By establishing a structured foundation for future model development, this work provides a roadmap to advance the integration of ecosystem services in life cycle impact assessment.

Keywords: life cycle impact assessment, ecosystem services, impact modeling, characterization

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1. Introduction

Nature is essential for human well-being and activities, which depend on the stock of natural assets collectively known as "natural capital" (IPBES 2019). Recognition of the significance of nature for humans led to the development of the concept of "ecosystem services" (ES), defined by Costanza et al. (1997) as the "benefits populations derive, directly or indirectly, from ecosystem functions" and highlighting the link between human well-being and ecosystem functioning within social-ecological systems (Costanza et al. 1997; McGinnis and Ostrom 2014). Natural Capital Accounting (NCA) methods, such as the Natural Capital Protocol (Natural Capital Coalition 2016) highlight not only the dependency of human activities on functioning ecosystems but also the influence of our activities on the ability of ecosystems to provide benefits to people.

Due to increasing environmental pressures, the ability of ecosystems to provide services is at risk, threatening human well-being (Millenium Ecosystem Assessment 2005; IPBES 2019) To prevent further decline, integrating ES into policy and decision making is crucial to support management practices that ensure the sustainability of these services (European Commission 2020)

Life cycle assessment (LCA) is a prominent tool for supporting sustainable decision making and policy development (Hellweg et al. 2023). This methodology assesses the potential environmental impacts of human activities on subjects that are important to society and need to be protected, referred to as Areas of Protection (AoP) (Schaubroeck and Rugani 2017; Verones et al. 2017). Current common AoP include human health, ecosystem quality, and natural resources (Huijbregts et al. 2017).

Although LCA scholars have acknowledged the societal importance of ES (Verones et al. 2017; Bulle et al. 2019), their operational integration in LCA remains limited (Vanderwilde and Newell 2021). However, such inclusion is essential to comprehensively capture "the hidden costs or benefits to society of [human activities] degrading or supporting functional ecosystems" (Othoniel et al. 2016).

Different approaches have been proposed for the integration of ES in LCA (Vanderwilde and Newell 2021; Rugani et al. 2023). However, a critical gap remains in the Life Cycle Impact Assessment (LCIA) phase, where ES concepts have not yet been fully incorporated into existing impact characterization models and indicators (Othoniel et al. 2016; Alejandre et al. 2019).

This study hypothesizes that damage-oriented ES assessment in LCIA is hindered by the absence of harmonized and consistent modeling guidelines and terminology.

To investigate the validity of this research hypothesis, this critical review seeks two specific objectives. The first objective is to **establish a foundational understanding of ES** to provide a knowledge base for evaluating existing LCIA ES models and guiding the development of future ones. This includes clarifying key concepts such as instrumental value (Arias-Arévalo et al. 2017, 2018; Schroeder 2024) and cascading models (Potschin-Young et al. 2018) . The second specific objective aims at **critically assessing existing proposals for integrating ES into LCIA at the light of this foundational**

understanding of ES. This involves reviewing and discussing proposed methods in relation to ES knowledge, identifying conceptual and terminological misalignments, methodological gaps, and areas for improvement.

The findings of this article aim to harmonize the development of LCIA models, fostering the integration of ES concepts into LCA and decision-making processes.

2. Methodology

This critical analysis is based on two distinct literature reviews to support a comprehensive understanding of ES and their integration into LCIA. The first review focused on foundational concepts within the ES field, while the second review critically assessed existing approaches to incorporating ES within LCIA.

The first literature review examined recognized publications in the field of ES. This review focused on key sources widely cited within the ES community. The foundational texts included The Millennium Ecosystem Assessment (MEA) (Millenium Ecosystem Assessment 2005), influential works by Daily (1997), De Groot et al. (2002), Potschin-Young et al. (2018) and Costanza et al. (1997, 2017), key reports from the Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services (IPBES 2019), and other significant frameworks and classification systems, such as the Common International Classification of Ecosystem Services (CICES) (Haines-Young et al. 2023). To expand this initial set of sources, relevant studies were identified through forward and backward citation tracking.

The second literature review critically examined the integration of ES in LCIA, with a focus on characterization models and methods. The search was performed in the Compendex and Google Scholar databases. The following keywords were used in various combinations to capture the relevant literature: ’ecosystem services’, ’ecosystem functions’, ’ecological functions’, ’life cycle impact assessment’, ’LCIA’, ’life cycle assessment’, ’LCA’, ’characterization’, ’impact’, and their variations. The initial search results were selected based on the relevance of the topic, specifically focusing on studies that discussed methods, frameworks, or models for incorporating ES into LCIA.

Foundational knowledge from the ES community derived from the first review was used to assess the methodological rigor and coherence of ES integration approaches in LCIA literature. This critical synthesis provided a foundation for recommendations on improving ES integration.

Section 3 provides the review of the foundational concepts of ES, while Section 4 examines the integration of ES within LCA and section 5 presents the results of the critical analysis of existing proposals to incorporate ES into LCA.

3. Fundamental understanding of ecosystem services

3.1. The CICES ecosystem services classification

Bringing together more than 1300 experts, the Millenium Ecosystem Assessment (2005) has catalyzed the interest of scientists and decision-makers in ES and proposed the first ES classification. Following this initiative, other classifications have been proposed (for example, Fisher et al. (2009; De Groot et al. (2010). The most common classifications include The Economics of Ecosystems and Biodiversity (TEEB) framework (TEEB 2012) and the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin 2018) which was developed in 2009 by the European Environment Agency. Based on the MEA and TEEB typologies, CICES has become a reference for the ES community (Maes et al. 2016; La Notte et al. 2017; Finisdore et al. 2020).

The CICES classification categorizes ES into 3 main sections: 1-provisioning ES (e.g. provisioning of water, timber, minerals, biomass, etc.), 2-regulation and maintenance ES (e.g. mediation of waste, erosion control, or hazard mitigation) and 3-cultural ES (e.g. recreational activities or spiritual interaction with natural system). These 3 ES categories rely on ecological processes, such as nutrient cycling, soil formation, or primary production, which were considered as supporting/intermediate services in the MEA and are currently defined as the "underpinning structures and processes that ultimately give rise to ecosystem services" in CICES.

Initially, "CICES has focused on defining [...] ecosystem services that depend on living systems" (Haines-Young et al. 2023). This exclusive biocentric focus has been criticized by some authors as it ignores the benefits offered by geodiversity ("diversity of geological structures and processes, including rocks and minerals; geomorphology, including land forms and topography; sediments and soils, including formation processes; and hydrology, including marine, surface, and subsurface waters" (Fox et al. 2020)), and overlooks synergies among biotic and abiotic domains (Gray 2012; Gray et al. 2013; Van Ree and van Beukering 2016; Baveye et al. 2018; Fox et al. 2020). "Geosystem services" can include the provision of rare-earth metal, water or the regulation of thermal flows, and are often referred to as "abiotic" ES (Gray, 2012; Fox et al., 2020). What is often referred to as "biotic" ES is largely dependent on biophysical processes (i.e., based on a combination of biotic and abiotic processes) and not only biological processes (Gray 2012; Fox et al. 2020; Frisk et al. 2022). In response to these criticisms, the latest version of CICES (v5.2) combines the "biophysical ecosystem services" and "geophysical services" sections within the same ES classification. Hence, each section of the v5.2 classification (provisioning, regulation & maintenance and cultural) includes bio- and geophysical services.

The three CICES categories of ES, segmented into divisions, groups, and classes, intend to represent the final services offered by nature used by the social-economic system. The CICES classification, even if more detailed than most of the other ES classifications, remains limited to relatively coarse levels of detail in terms of service provided. For example, the provisioning ES of mineral substances in CICES is limited to the provisioning of substances 1-for nutritional purposes, 2-for material purposes, and 3-as energy source (codes 4.2.1.1, 4.2.1.2, 4.2.1.3 in CICES v5.2). Within each of these categories, flow of ES can be used for more specific final purposes (e.g., minerals ingested by humans like salt, or minerals used as crop fertilizers), which are not detailed for practical reasons. For specific, context-dependent assessments, practitioners may further specify the service or use refined classifications such as Whiting et al. (2022), which detail the services that can be provided through the provision of natural resources.

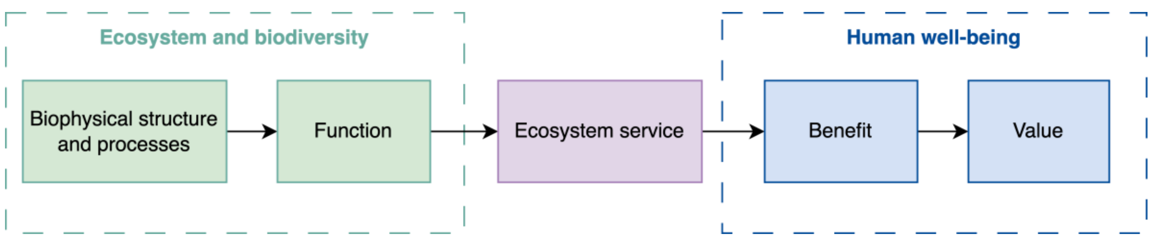

Figure 1: ES cascade model, adapted from De Groot et al. (2010).

3.2. Ecosystem services frameworks

Like most ES classifications, CICES is based on the "cascade model" proposed by Haines-Young and Potschin (2010)The latter illustrates the linkages between an ecosystem's structure and processes which, through ecosystem functions, provide the support for human well-being (De Groot et al. 2010; Haines-Young and Potschin 2010; Kandziora et al. 2013; Burkhard et al. 2014; La Notte et al. 2017) (Figure Figure 1). The structure of an ecosystem depends on its biotic and abiotic components as well as its physical features (e.g., climate or nutrient distribution) while ecosystem processes are defined as the physical, chemical and biological interactions linking organisms and their environment (e.g., food web dynamics or nutrient cycling). The structure and processes of an ecosystem underpin their **functions**, defined in the TEEB (2010) as the "potential that ecosystems have to deliver a service" (De Groot et al., 2010). Burkhard et al. (2012) define the **ES potential** as the "hypothetical maximum yield of selected ecosystem service". The **ES flow** is the fraction of this potential used by humans. The notion of **demand** by humans is a prerequisite for the existence of an ES which does "not exist in isolation from people's needs" (Haines-Young and Potschin, 2010). The contribution of an ES to human well-being is accounted for as a benefit, and a value can be assigned to it.

Spangenberg et al. (2014) highlighted the societal conditions required for certain ecosystem services to exist, such as legal frameworks that allow the extraction of natural resources, including wood harvesting and mineral extraction. Kalt et al. (2019) proposed an energy service cascade which incorporates both the "socio-economic system's structures and processes" alongside the "biophysical structure and processes", building on the ES cascade model. These frameworks underscore the importance of socio-economic system's structures and processes in supporting certain ES.

3.3. Sustainable management of ecosystem services

The balance between ES supply and demand, called the "supply-demand" relationship, is crucial for sustainable management (Burkhard et al. 2012; Zhang et al. 2024). This concept emerged alongside the idea of ecological carrying capacity (Zhang et al., 2024), defined as "the maximum population level that a given environment can support given finite resources" (del Monte-Luna et al. 2004).

An ES flow is considered sustainable when it meets demand without compromising future provision (Villamagna et al. 2013). In contrast, it becomes unsustainable when current capacity cannot meet demand or when fulfilling demand reduces other services or the future availability of the same service. Therefore, ES sustainability declines when demand increases or when the ecosystem's capacity to provide services (that is, ES supply) decreases. The next subsection discusses changes in the ES supply.

3.4. Changes in ES supply

Various drivers of change can affect the functioning of ecosystems and their ability to provide ecosystem services (IPBES 2019). One key driver is land use change, which directly alters ecosystems and their capacity to provide services by impacting essential ecological processes such as energy flows, soil erosion and biogeochemical cycles

(Burkhard et al. 2012; Fu et al. 2015; Hasan et al. 2020). Other factors, such as changes in sea use, resource extraction, pollution, invasive alien species, and climate change, can also influence the delivery of ES (Hevia et al. 2017; IPBES 2019).

The effects of these drivers are closely related to the ecosystem's *resilience*, defined as its capacity to absorb disturbances while maintaining essential functions and structures (del Monte-Luna et al., 2004). Regulating services within the ecosystem play a crucial role in mitigating the impacts of ecological pressures (Villamagna et al., 2013). For long-term sustainability, the demand for goods and services must remain within the ecosystem's ecological *carrying capacity* (Goodland and Daly 1996).

3.5. Different dimensions of the values of nature

Valuation debates have recently evolved towards acknowledging three different value dimensions of nature to describe the ways in which nature and ecosystems are important to people. To account for this pluralism of values, scholars distinguish between intrinsic, instrumental, and relational values (Arias-Arévalo et al. 2017; Kadykalo et al. 2019; Pereira et al. 2020; Himes et al. 2024) . Value types refer to different aspects of the importance of an entity. However, an entity (e.g. an object, a place, an animal, an ecosystem, etc.) can support different types of value. **Intrinsic** values of nature are described as values inherent to a natural entity simply by virtue of its existence and independently of its utility for human societies (Hargrove 1992; Turner et al. 2003; Arias-Arévalo et al. 2017). **Instrumental** values refer to entities that are useful to people and are described as "means to achieve human ends or satisfy human preferences" (Arias-Arévalo et al. 2017; Himes and Muraca 2018; Hill et al. 2021; Himes et al. 2024). They are strongly and explicitly associated with anthropocentrism and utilitarianism (Himes et al., 2024). Unlike its intrinsic value, the instrumental value of an entity is substitutable (Himes and Muraca, 2018): other entities can be used to achieve the same ends.

The concept of **relational** values seeks to move beyond the traditional dichotomy of intrinsic and instrumental values, which has long dominated the discourse on sustainability (Pascual et al. 2023). Relational values refer to meaningful and non-substituted relationships between humans and nature, such as the relations of responsibility and care toward nature, eudemonism or place-based value (Chan et al. 2018; Himes and Muraca 2018; Hill et al. 2021). Himes et al. (2024) defined relational values as "values of meaningful and often reciprocal human relationships -beyond means to an end- with nature (often specified as a particular landscape, place, species, forest, etc.) and among people through nature".

Defined from an anthropocentric point of view, the concept of ES supports instrumental values only. However, in response to the emergence of the notion of relational values, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has broadened the concept of ES by introducing the concept of "nature’s contributions to people" (NCP) (IPBES 2019; Hill et al. 2021). The NCP concept is broader and more inclusive than ES and acknowledges that our appreciation of nature’s benefits to people is multidimensional and shaped by context, culture and worldviews, including indigenous and local knowledge (Díaz et al. 2018; Hill et al. 2021; Marcos et al. 2021; Pascual et al. 2023).

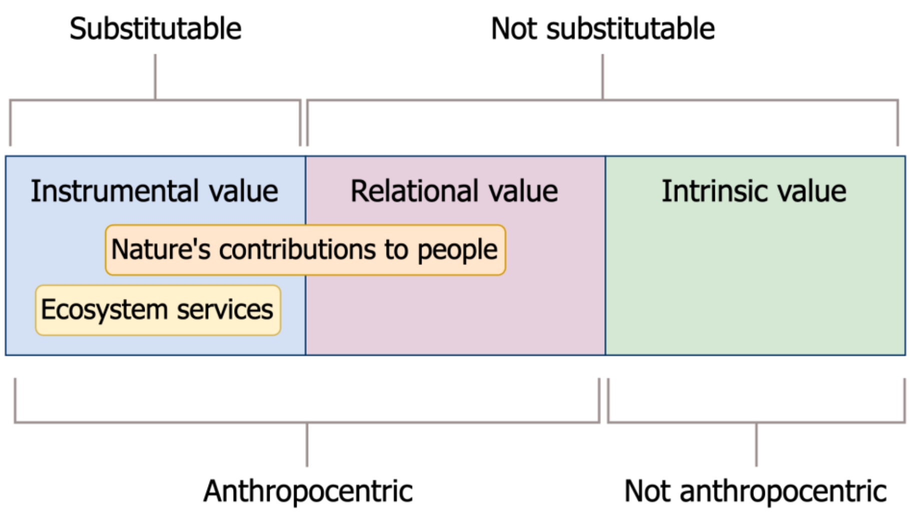

Figure 2: Domains of values of nature. Adapted from Himes and Muraca (2018). The figure positions ES and nature’s contributions to people in the scope of this paper. Nature’s contribution to people extends the scope of ES to encompass relational values and acknowledge more diverse conceptualizations of human-nature relationships

To avoid any ambiguity in terminology, this article uses the term "ecosystem services" sensu stricto, i.e. restricted to the instrumental value of nature only, and the term "nature’s contributions to people" as a concept encompassing both the instrumental and relational value of nature (see Figure 2).

3.6. Instrumental use and non-use values

Instrumental values (and therefore ES) encompass both the *use value* and the *non-use value* (Turner et al. 2003; Himes et al. 2024) (see *Figure 3*). Use value encompasses actual value (current use) and option value (expectation of future use).

Actual value is linked to both the direct and indirect use of nature. Direct use can be consumptive (e.g. collection of berries) or nonconsumptive (e.g. birdwatching). Indirect use value mostly relies on regulation and maintenance of ecosystem functions, which people can benefit from without interacting directly with ecosystems.

Non-use values are derived from knowledge of the continued existence of nature (Davidson 2013). They encompass the warm glow value (valuing benefits for other people or future generations) and the existence value (satisfaction with knowing that nature exists).

A common mistake lays in the confusion between existence and intrinsic values. The existence value is "related to the satisfaction that people may derive from the mere knowledge that nature exists" and can be classified as a cultural ES (Davidson, 2013). However, the concept of intrinsic value breaks away from an anthropocentric judgment and "lie[s] outside the scope of the wide palette of ES" (Davidson, 2013).

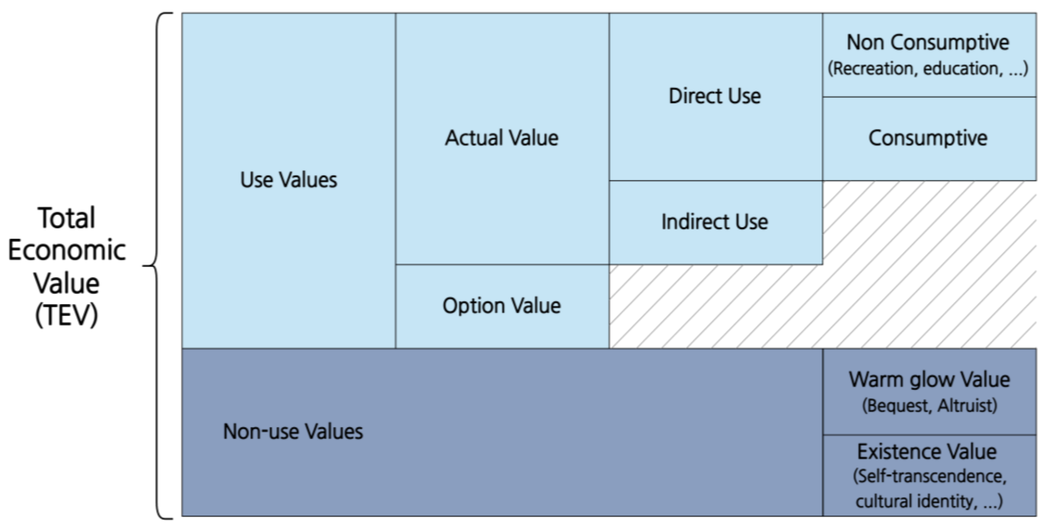

Figure 3 : Total Economic Value (TEV) framework and examples of valuation methods that could be used to assess it (partially or totally), adapted from Davidson (2013). Valuation methods are grouped by families. hatched bars: stated preferences methods, doted bars: revealed preferences methods, cross hatched bars: market-based methods.

4. ES in life cycle impact assessment

4.1. Life Cycle Assessment

LCA is a prominent methodology designed to support sustainable decision-making by assessing the potential environmental impacts of human activities across all stages of a system’s life cycle (International Organization for Standardization 2006).

LCA operates in 4 phases, namely (i) the definition of the goal and scope, (ii) the life cycle inventory (LCI), (iii) the life cycle impact assessment (LCIA), and (iv) the interpretation of results (ISO 2006). Within the system boundary, LCA distinguishes between foreground processes, which are directly influenced or controlled by the decision-maker, and background processes, which are more generic and support activities beyond direct control (Frischknecht 1998)

During the LCI phase, the exchanges of material or energy flows between the product system and the ecosphere, called elementary flows, are compiled in the inventory (International Organization for Standardization 2006). The LCIA phase aims at translating the inventory flows into potential environmental impacts. In this respect, characterization factors (CFs) are multiplied with the different elementary flows to calculate impact scores. The latter are combined to provide an environmental impact profile. Impact indicators are distinguished at the "problem" level (with an indicator chosen anywhere along the cause effect chain and typically expressed in equivalent of a reference elementary flow such as global warming, in $kgCO_2e$) and at the "damage" level (with an indicator chosen at the end of the cause effect chain measuring the damage on an AoP and expressed in absolute values, e.g., climate change-driven potential impact on human health, in DALYs). Notwithstanding impact scores at damage level encompass a higher level of model uncertainty (Bare et al. 2000), they might provide more relevant and tangible information to decision-makers (Bare et al. 2000; Torkayesh et al. 2022).

Damage-oriented LCIA methodologies (compiling several characterization models) such as IMPACT World+ (Bulle et al. 2019), ReCiPe (Huijbregts et al. 2017)or LC-Impact (Verones et al. 2020) enable aggregating different damage-oriented impact categories to different AoP. These AoP represent entities that are important for society and that LCA aims to safeguard (Schaubroeck and Rugani 2017; Verones et al. 2017).

Historically, three AoP have been considered by the LCIA community: human health, ecosystem quality and natural resources. These AoP are central to well-known impact assessment methodologies such as LC-Impact (Verones et al., 2020), IMPACT2002+ (Jolliet et al. 2003) or ReCiPe (Huijbregts et al., 2017). However, different ethical perspectives on what should be sustained can lead to the development of different AoP (Schaubroeck and Rugani, 2017). The choice of what an AoP entails - hence what society deems worth protecting - is based on value assumptions, ethical perspectives, and epistemology of the research community and thus open to debate (Freidberg 2018; Pascual et al. 2021).

4.2. Integrating ES into Life Cycle Assessment

Despite shared sustainability ambitions, the LCA and ES communities have grown separately (Vanderwilde and Newell, 2021). However, in the past decade, efforts have been made to integrate ES into LCA (Zhang et al. 2010; Koellner and Geyer 2013; Koellner et al. 2013; Maia de Souza et al. 2018; Vanderwilde and Newell 2021; De Luca Peña et al. 2022; Rugani et al. 2022; Heitbrink 2024).

De Luca Peña et al. (2022) define three *categories* of research approaches for integrating ES and LCA methodologies:

- the "post-analysis" category involves conducting ES and LCA analyses independently, with results interpreted in parallel.
- the "integration through combination of results" group entails conducting the analyses separately but combining their results prior to interpretation.
- the "integration through the complementation of a driving method" approach designates one methodology -either LCA or ES- as the primary framework, enhanced by tools or insights from the other. In this latter category, LCA is the most frequent driving method.

Within this last category, the classification of Rugani et al. (2023) further distinguishes research efforts into four *paradigms* : 1- the inclusion of ES as part of the functional unit of a system, 2- the inclusion of human activities' use of ES through the life cycle inventory, 3- the inclusion of regulation and maintenance services as a means of mitigating environmental impacts, and 4- the potential impacts of human activities on ecosystems' capacity to provide ES (assessed either at the problem or damage level).

Rugani et al. (2023)'s paradigms are not mutually exclusive and can be combined. They suggest that integrating ES into LCA can be relevant at every phase of the assessment. Moreover, Rugani et al. (2019, 2023) suggest that the best approach to integrate ES flows might depend on their type; some ES flows should serve as functional units (e.g., provisioning services like crops), while others should be considered elementary flows in the LCI stage (e.g., provisioning services such as freshwater or biomass). Certain ES flows should be assessed in LCIA, such as the impact of human activities on carbon sequestration. Finally, some ES flows such as aesthetic services, ecological habitats that support species should be considered separately, as "natural capital components to be protected".

Building on the classifications of De Luca Peña et al. (2022) and Rugani et al. (2022), a third exhaustive classification of approaches for ES-LCA integration is formulated in *Table 1*.

Table 1: Classification of approaches to integrate ES and LCA derived from the combination and enhancement of De Luca Peña et al. (2022)’s and Rugani et al. (2023)’s classifications. n is the number of propositions using or mentioning the approach for an ES-LCA integration among the 43 reviewed studies. One study can use or mention several approaches. See ESM1 for additional information.

Classification by De Luca Peña et al. (2022)	**Classification by Rugani et al. (2023)**	**Classification in this work**	n
LCA as "driving method”	In the Functional Unit	In the Functional Unit	2
		Considering multi-functionality	3
	In the inventory	In the inventory	19
	Regulation and maintenance services as a means of mitigating impacts	Impact mitigation/avoidance by regulating and maintenance ES	2
	Potential impacts on ecosystem’s capacity to provide services	Impacts on ecosystems’ capacity to provide ES : stress	16
		Impacts on ecosystems’ capacity to provide ES : transformation	17
"Post-analysis": LCA and ES performed independently		"Post-analysis": LCA and ES performed independently	1
"Combination" of damage-oriented results		"Combination" of damage-oriented results	6

This revised classification refines some of the categories of Rugani et al. (2022). The "integration through the functional unit" approach is further divided into two parts : the formulation of a functional unit which can be provided by an ES (e.g. FU="Capture 1 ton of CO_{2eq}"), and the treatment of an ES via the issue of multi-functionality (e.g. in the assessment of an agro-ecosystem providing food among other ES (Boone et al. 2019)). Similarly, building on Hardaker et al. (2022), the "potential impacts on ecosystems’ capacity to provide services" approach is divided into two parts in *Table 1*: the impacts due (1) to stress to and (2) to transformation of ecosystems.

The revised classification developed in this paper was used to position studies proposing to integrate ES in LCA. The detailed result can be found in the SI.

In response to the methodological challenges involved in the "LCA as driving method" approach, Rugani and colleagues (2022) suggested "intertwining" LCA and ES tools in a framework where inventories and midpoint indicators are connected, while the goal and scope phase, the endpoint-oriented results and the interpretation phase are shared. This perspective aligns with the "integration through combination of results" approach proposed by De Luca Peña et al. (2022), which advocates for moving beyond the exclusive reliance on LCA to deliver comprehensive assessments that encompass ecological dynamics, ES benefits, and environmental impacts. The intertwining of ES and LCA tools and the application of ES assessment at the foreground level is particularly interesting for product systems such as Nature-based solutions (Larrey-lassalle et al. 2022; Rugani et al. 2022; Alshehri et al. 2023; Babí Almenar et al. 2023) such as revegetation. In this context, the joint use of tools allows underlining both the degree to which the system benefits from (or depend on) ES and the extent of its environmental impact throughout its life cycle.

Rugani et al. (2022)'s suggestion to interweave ES and LCA tools neither eliminate nor are incompatible with the intention to further develop life cycle characterization models and factors assessing potential impacts of various drivers on ecosystems' capacity to provide services.

4.3. Rationale for integrating ES into Life Cycle Impact Assessment

The integration of ES concepts within LCIA can promote the attention of decision makers toward potential impacts on ES and reveal the "hidden costs or benefits to society of the LCI flows degrading or supporting functional ecosystems" (Othoniel et al. 2019). Conversely, the current exclusion of ES from LCA impact profiles constitutes a significant blind spot in LCA impact results (Rugani et al. 2022). Since LCA aims to provide a comprehensive assessment of environmental impacts, this omission can distort sustainability-driven decision-making. and encouraging decisions that inadvertently compromise ecosystems' ability to provide essential services (Liu et al. 2018).

5. A critical review on the proposals to integrate ES into LCIA

5.1. *Proposals to integrate ES into LCIA as an Area of Protection*

Methodological advances and innovations in LCIA models have led the UNEP-SETAC Life Cycle Initiative to review the LCIA framework and its AoP (Verones et al., 2017) (**Erreur! Source du renvoi introuvable.**).

Table 2: Classification of AoP depending on the type of object deserving protection and the social value assigned to it, as per Verones et al. (2017). Verones et al. (2017) distinguish intrinsic, instrumental and cultural values, which can emerge from human and human-made systems or from natural systems. In this work, we suggest to replace cultural values by relational values.*

Type of objects \ Type of value	**Intrinsic values**	**Instrumental values**	**Cultural values***
Human and human-made systems	Human health	Socio-economic assets	Cultural Heritage
Natural systems	Ecosystem Quality	Natural resources & ecosystem services	Natural Heritage

The working group identified two categories of entities deserving protection that need to be distinguished in LCIA : entities pertaining to (i) human and human-made systems and to (ii) natural systems. Within these categories, the authors consider that entities can have "intrinsic", "instrumental", and "cultural" values.

The expression "cultural values" in Verones et al. (2017)’s presents limitations. The term “cultural” diverges from the value classifications introduced in Section 3.5. Besides, current ES classifications categorize “cultural” ES under the *instrumental* value of natural systems (Haines-Young et al. 2023). In this context, using the term "cultural" may create confusion. Conversely, the emerging concept of "relational values" is more suitable for capturing the full spectrum of values and is used in the remainder.

Verones and colleagues’ double-entry table suggests creating an AoP for each category of entity and corresponding type of value. The "human health" and the "ecosystem quality" are described as entities with intrinsic value that belong to the "human/human-made" and "natural" systems, respectively. Similarly, Verones et al. (2017) consider "cultural heritage" and "natural heritage" as AoP representing entities with relational values, pertaining to "human/human-made" and "natural systems", respectively. Finally, Verones et al. (2017) define "Socio-economic assets" and "Natural resources and ES" as entities with instrumental values pertaining to the "human/human-made" and "natural" systems, respectively. Despite limitations in Verones et al. (2017)'s classification of AoP (further discussed in section A of the Supplementary Information) it is adopted in the remainder of our analysis.

Verones et al. (2017)'s proposal to develop a "Natural resources & Ecosystem services" AoP reflects the growing interest among LCA practitioners and LCIA model developers to include information on ES and instrumental values in LCA endpoint-oriented impact profiles. This increasing interest is also illustrated by the creation of a specific task force group within the 3rd phase of the Global Life Cycle Impact Assessment Method (GLAM) initiative (Life Cycle Initiative 2021). However, only a few studies have provided operational ES characterization models and factors at the damage level. The following paragraphs explore the limitations linked to the development of the latter.

5.2. The LCIA community treats Natural Resources and provisioning Ecosystem Services separately despite their interconnectedness

The LCIA community has largely overlooked the inherent connection between natural resources and provisioning ES. This disconnect may stem, in part, from historical ES classifications that focused primarily on services derived from biotic processes and excluded geo-resources;

the integration of market-valued natural resources into LCIA frameworks preceded the inclusion of broader ES categories (UNEP/SETAC 2019).

Research studies have been carried out on abiotic resources such as water (Pradinaud et al. 2019), metal or mineral resources (Sonderegger et al. 2020a; Charpentier Poncelet et al. 2021; Pradel et al. 2021; Greffe et al. 2022; Ardente et al. 2023; Beylot et al. 2024), soil (Sonderegger et al. 2020b), marine surfaces (Taelman et al. 2014), or biotic resources (Crenna et al. 2018) such as wood (Odppes et al. 2021) or halieutic resources (Langlois et al. 2014; Luong et al. 2020; Hélias et al. 2023; Stanford-Clark et al. 2024).

Although these natural resources are provided by biophysical (e.g. timber or fish provisioning) or geophysical provisioning ES (e.g. mineral resources), most studies do not explicitly refer to ES concepts. The absence of efforts to clarify the links between resources and ES within the LCIA community has led to the parallel development of models, tailored for natural resources (Sonderegger et al. 2017) on one hand and for other ES (Rugani et al. 2019) on the other.

These siloed development efforts have resulted in conceptual and terminological inconsistencies. Within the resources-oriented LCIA community, the term "functionality of natural resources" is used to refer to the instrumental value that a resource provides to society, whereas in the ES community, "ecological functions" describe the biophysical processes that sustain ecosystems. Moreover, both ES and resources-oriented LCIA communities use the term "service" to denote the benefits that ecosystems or resources provide to society, referred to as "ES" and "resources services", respectively. However, in practice, the level of detail in these services differs. CICES classifications delineate resources at the class level (e.g., CICES v5.2 class 4.2.1.2: *mineral substances used for material purposes*), while resources-oriented characterization models in LCIA often extend beyond resource supply to specify the societal service provided by the resource (e.g., *materials used for mobility*, such as aluminum used for vehicles). Harmonization and clarification of the level of services to be considered in the characterization models could clear up this ambiguity. Research efforts reconciling CICES with resources' services such as Whiting et al. (2022)'s classification could be explored to address these limitations.

5.3. Several proposals for integrating ES into AoP have been published but many lack clarity and consistency

Although Verones et al. (2017)'s social value classification suggests a common AoP for natural resources and ES (see **Erreur! Source du renvoi introuvable.**), their conceptual LCIA framework does not clearly differentiate between the two. In Figure 1 of their article, resources and ES are depicted separately, reinforcing the distinction discussed in section 5.2. Following Verones et al. (2017)'s recommendation, Bulle et al. (2019) implemented a "Resources and Ecosystem Services" AoP in their LCIA methodology. However, the terminology used for this AoP has caused confusion: while it was intended to represent "the services provided by both resources and ecosystems", it has often been interpreted as "resources" (on the one hand) and "the services provided by ecosystems" (on the other).

Other scholars offered different approaches for integrating ES into AoP. In their theoretical LCIA framework, Hardaker et al. (2022) proposed separating ES completely from natural resources, recommending a standalone "ecosystem services" AoP to accompany existing AoP for human health, ecosystem quality, and natural resources. Their ES-specific AoP combines two aspects: (1) the use of ES supply (to what extent do

ES mitigate midpoint impacts generated by the studied system?) and (2) the change of ES supply (to what extent does the studied system change ecosystems' capacity to provide ES?). These proposals represent innovative contributions to LCA by emphasizing human *dependence* on natural capital and ES, an aspect typically overlooked in traditional LCA impact profiles and crucial for the ES community (Natural Capital Coalition, 2016; IPBES, 2019). However, one should keep in mind that the notion of human reliance on natural systems does not correspond to an assessment of the impact of the product system on the environment, but rather to an assessment of the dependence of the product system on the environment. Hence, it is debatable to position this as an impact pathway in the impact assessment framework.

Taelman et al. (2022) offered a different perspective by proposing to integrate ES into existing AoP; the "Natural Resources" AoP encompasses provisioning ES, the "Ecosystem Quality" AoP is broadened to include Regulating and Maintenance ES, and the "Human health and well-being" AoP "covers both human health impacts and cultural ES". Similarly, Rugani et al. (2019) establish links between *Provisioning ES* and the Natural Resources and Ecosystem Quality AoP, and between both *Regulation and maintenance*, and *Cultural ES* with Human well-being and Ecosystem Quality AoP. These works are among the few that highlight the similarities between provisioning ES and natural resources (see section 5.2).

However, by separating ES between AoP, Taelman et al. (2023)'s proposal risks overlooking the intricate links between ES flows and the broader AoP of human health and ecosystem quality (see section 5.7), a limitation identified by the authors. Moreover, this proposal diverges from Verones et al.'s approach, which advocates for separate AoP for each system type, based on its specific value type .

5.4. Modeling purposes for Natural resources (which are provisioning ES) lack transparency and coherence

The modeling "purposes" behind LCIA characterization models are shaped by what developers believe should be protected (Freidberg 2018). For natural resources these purposes vary widely (Dewulf et al. 2015)

The diversity of natural resources, from minerals, metals, fossil fuels and air components to land and water, combined with their typological differences (stocks, funds, and flows), poses a significant challenge to harmonize impact modeling across different approaches (Sonderegger et al. 2017; Pradinaud et al. 2019). However, beyond the characteristics of the resources, the aspect of the resource that is intended to be protected plays a critical role in shaping the characterization models.

The various modeling methods that have been proposed (Sonderegger et al., 2017) often address different research questions, which in turn have led to the development of different impact assessment approaches. For example, some LCIA models aim to assess the potential impacts on the (reduced) availability of natural resources for future generations (e.g. Pradinaud et al. (2019)), their stock in the earth's crust (van Oers and Guinée 2016; van Oers et al. 2020), the efforts required to make this stock accessible (Vieira et al. 2017), their exergy content (Dewulf et al. 2007) or cumulative energy demand (Frischknecht et al. 2007), among other indicators.

Dewulf et al. (2015) explored the diverse interpretations of the AoP specific to natural resources, identifying three key safeguard subjects: (1) the Asset of Natural Resources, (2) the Provisioning Capacity of Natural Resources, and (3) the Role of Natural Resources in Global Functions. Similarly, Berger et al. (2020) noted that the selection of a characterization model often depends on its intended application, underscoring the variety of perspectives within the field.

However, the recent consensus of Verones et al. (2017) advocates that characterization models should focus on the *instrumental value* of natural resources, particularly in terms of the services they provide to humans. This shift is reflected in the AoP "Resources and Ecosystem Services" of the IMPACT World+ methodology, where models are increasingly expected to capture the functionality of resources for human society (Bulle et al., 2019). However, earlier models often failed to incorporate this functional perspective. For example, early mineral resource-oriented methods focus on resource extraction (Berger et al. 2020). However, the extraction process fails to address a loss of resources' service per se : a resource extracted from the lithosphere and used in the technosphere can still provide instrumental value for human society if recycled (Greffe et al. 2022). In contrast, the degradation and dissipation of materials threaten the functionality of natural resources (Charpentier Poncelet et al. 2021, 2022; Graedel and Miatto 2022; Greffe et al. 2022). In response, newer characterization models are evolving to assess the potential loss of services due to resource dissipation (De Bruille 2014; Greffe et al. 2022; Beylot et al. 2024)

5.5. Assessment of ES (beyond natural resources) lacks harmonization, completeness and guidelines

In this section, the term "ES" is used as adopted by the LCIA community, which distinguishes between ES and services provided by natural resources, although the latter derive from provisioning ES.

Despite more than a decade of attempts to integrate ES concepts into LCA, their integration at the LCIA level remains in its infancy (Hardaker et al. 2022; Rugani et al. 2022).

Our literature review identifies two schools of thought regarding the assessment of ES in LCA: i) researchers focusing on grasping the ES supply-demand, and ii) researchers focusing on modeling the impacts of human activities on ecosystems' capacity to provide services. Although these approaches differ in perspective, their proposals partly overlap.

5.5.1. i) Academics interested in the ES "supply-demand" relationship

Although conventional LCA primarily focuses on assessing potential environmental impacts, the ES community places a significant emphasis on the dependence of human activities on natural capital and on the ES supply-demand relationship (Burkhard et al. 2012; Xue and Bakshi 2022). Some LCA researchers have proposed theoretical frameworks and models that align with this philosophy (Bakshi and Small 2011; Rugani et al. 2019; Liu et al. 2020; Hardaker et al. 2022; Xue and Bakshi 2022).

The approaches proposed by Liu et al. (2018)) and Liu et al. (2020) aim to extend the classic steps of LCA to incorporate the notions of *demand* and *supply* of ES at different spatial scales. Their proposals build on Bakshi and Small (2011)'s Techno Ecological Synergies (TES) framework aiming at "understanding and designing synergies between technological and ecological systems". Through this framework, the authors reinterpret the environmental intervention matrix (used in conventional LCA to compile exchanges between the ecosphere and the technosphere) into a *demand* matrix. In the latter, flows of resources are referred to as demand for provisioning ES, and flows of substances emissions are referred to as demand for regulation and maintenance services. By comparing demand and supply flows for each ES, Liu et al. (2018) and Liu et al. (2020) derive sustainability scores: if demand induced by the activity exceeds the supply potential, the activity is deemed unsustainable.

The demand for regulation and maintenance services reflects the third paradigm of Rugani et al. (2022) (see section 4.2), which posits that by exerting pressures on ecosystems, human activities generate a form of *demand* for such ES to mitigate the damage they induce. Similarly, Hardaker et al. (2022) emphasize the demand for regulating and maintenance services by introducing an impact indicator that quantifies “the portion of global ES used to mitigate impacts at the problem level”.

Demand for ES : straddling between the inventory and impact assessment. The interpretation of Hardaker et al. (2022)’s indicator as an "impact" is debatable. This indicator differs from the impact indicators conventionally used in LCIA that reflect a change in the state of the system as the result of a cause-effect chain (Hauschild, Michael Z and Huijbregts 2015). Rather, one could interpret it as a descriptor or the state of a system that accounts for the quantity of services it provides. In this context, proposals to quantify the demand for ES could be regarded as part of the LCI phase rather than of the LCIA phase.

This ambiguity on the positioning of demand-supply approaches within the LCA framework can also be found in Liu et al. (2018) and Liu et al. (2020). The authors’ reinterpretation of the environmental matrix into a "demand matrix" reinforces the link between their approach and the LCI phase. However, quantifying the demand for regulation and maintenance services requires an upstream impact assessment step. It is therefore difficult to classify this approach as part of the LCI or the LCIA phases.

From a LCIA perspective, their conceptualization of demand can be interpreted in two different ways depending on the type of ES; the demand for provisioning ES reflects the consumption of natural resources, which is captured at the LCI level, while the demand for regulation and maintenance services can be viewed as a positive impact associated with the mitigation of the impacts of the product systems and embedded into the impact assessment phase.

The ES demand-supply approach to assess "absolute environmental sustainability". Some LCA scholars have emphasized the connections between ES concepts, including the relationship between ES supply and demand, and absolute environmental sustainability (AES) metrics (Liu et al. 2018; Hardaker et al. 2022; Xue and Bakshi 2022).

Several LCA-based AES assessment methods have been proposed to "evaluate whether an anthropogenic system can be considered environmentally sustainable in an absolute sense" (Bjorn et al. 2020). In these methods, the impact of the anthropogenic system is compared to its "assigned carrying capacity". These estimates of carrying capacity are often derived from the planetary boundary framework (Bjørn et al. 2016), which defines a "safe operating space" constrained by nine ecological boundaries (Steffen et al. 2015).

Some scholars have emphasized the relevance of ES concepts in promoting AES. Hardaker et al. (2022) argue that the integration of ES in LCA methodologies enables the evaluation of how the environmental burdens of product systems contribute to the encroaching on planetary boundaries. In the same perspective, other researchers propose to revise AES metrics based on ES. Through their TES framework, Bakshi et al. (2015)and Liu and Bakshi (2018) quantify AES by comparing "human demand for ES (emissions, resource use) with natural supply (carrying capacity of ecosystems)" (Xue and Bakshi, 2022). Unlike planetary boundary-based AES methods, which use downscaling to estimate local carrying capacity, the TES-LCA framework relies on biophysical ES models to estimate ES supply, "analogous to safe operating space", at different scales.

Challenges on the application of the supply-demand approaches. The quantification of the demand for regulation and maintenance services to mitigate potential impact presupposes an upstream impact assessment step, which in turn implies that a relevant impact category has already been selected during the goal and scope definition phase of LCA. Such an approach presents a challenge as at that early stage there is often no prior knowledge of which ES will be involved. The implementation of demand-supply approaches could eventually build on the iterative nature of LCA ; Initial conclusions drawn from the interpretation phase could inform refinements to the goal and scope definition, leading to subsequent refinements in ES integration in both the LCI and LCIA phases.

The methodologies proposed by Liu et al. (2018), Liu et al. (2020) and Rugani et al. (2022) involve expanding LCI flows to include the ES demand. Despite the conceptual advances of their proposals, the latter remain difficult to implement across an entire product life cycle the current structure and scope of LCA, which requires analyzing the environmental performance of background systems with unknown or globally distributed locations, do not allow for an explicit representation of the demand for ES mitigating these systems' impacts. Achieving this would require high-resolution regionalization. In practice, such flows can only be assessed at the foreground level, and Liu et al. (2018)'s approach remains constrained by the current capabilities of LCA software and LCI databases.

To answer similar challenges and account for both foreground and background processes within their "mitigation indicator", Hardaker et al. (2022) propose developing a database of regionalized characterization factors at various spatial scales, while acknowledging the challenges this proposal involves.

5.5.2. ii) Impacts on ecosystems' capacity to provide ES.

A second school of thought for the assessment of ES in LCA follows a more classic LCIA perspective and aims at modeling potential impacts on ecosystems' capacity to provide services.

In parallel with their "mitigation" indicator introduced earlier, Hardaker et al. (2022) proposed a second indicator to capture the impact of a product system in terms of "changes in ES supply",reflecting two impact pathways.

The first pathway highlights how ecosystem transformation, primarily driven by land use changes, alters its structure and functions and therefore threatens its capacity to provide services (Hardaker et al., 2022). This impact pathway aligns closely with common ES LCIA models, which are primarily designed to assess the potential impacts of land use.

The second pathway captures changes in an ecosystem's structure and functions caused by other environmental pressures -such as the emission of substances into soil, air, or water- "exceeding the carrying capacity of ecosystems to assimilate the emissions that cause these problems" (Hardaker et al., 2022).

The different proposals to integrate ES in LCIA have been developed based on one or both of these pathways. Vanderwilde and Newell (2021) classified these research efforts into three clusters: a) the land use cluster, b) the biodiversity and ES cluster and c) the dynamic modeling. The following paragraphs discuss the latter.

The *land use cluster* gathers characterization models and factors developed to assess the potential impacts on ES due to Land Use and Land Use Change (LULUC), linking land use to changes in ecosystem functions (Koellner and Geyer 2013; Koellner et al. 2013). The UNEP/SETAC LULCIA taskforce has laid the foundations for these approaches by developing modeling guidelines for the assessment of land use impact pathways (Milà i Canals et al. 2007; Koellner et al. 2013), helping conceptualizing the relationships between ecosystem functions, biodiversity and ES. Subsequently, other models have been developed based on these models and guidelines or to improve the characterization of land use (Cao et al. 2015; Bos et al. 2016). These land use impact-oriented models have the advantage of being operational in relation to existing inventory data (Rugani et al. 2022).

However, current land use characterization practices have significant limitations. First, they may not adequately account for the multitude of pressures that influence the capacity of the ecosystem to provide ES (Alejandre et al., 2019). Since land use is

commonly employed as a proxy for ecosystem transformations, there is also a risk of double-counting if future models incorporate additional pressure pathways (e.g. emission of substances into soil) that are not currently considered. Second, the representativeness of land use impacts is constrained by the scale and resolution of available inventory data (Othoniel et al., 2016; Rugani et al., 2022). Most ES depend on regional and local ecosystem conditions, which are often poorly captured by generalized land use categories in current LCI databases. Research initiatives aimed at refining inventory data resolution, such as Regioinvent (Agez 2025), offer promising pathways for enhancing the accuracy of land use inventory flows and, consequently, their relevance for ES assessments.
Furthermore, ES supply is time dependent (Othoniel et al., 2016). The ability to capture ES dynamics through land use flows is likely hindered by the seasonality of certain regulation and maintenance ES, which do not necessarily follow the same temporal patterns as technosphere flows modeled within the same dataset. Finally, ES are often provided through bundles (Saidi and Spray 2018), and existing models for land use impact assessment are not adequate to analyze human influence on the capacity of ecosystems to deliver multiple ES (Othoniel et al., 2016; Hardaker et al., 2022).

The *biodiversity and ES cluster* brings together methods establishing a relationship between biodiversity, ecosystem functions and ES (see section 5.7). Among these proposals, the framework proposed by Oginah et al. (2023) links the impacts on structural biodiversity to ES through changes in functional biodiversity. Following this framework, Li et al. (2023)) proposed a characterization framework linking microplastic and nanoplastic emissions to potential impacts on soil ES. These literature studies are promising attempts to capture ES concepts. However, the limited number of even very recent studies belonging to this cluster reflects a high degree of complexity and sophistication potentially associated with the approach.

The *dynamic modeling group* encompasses proposals to adopt dynamic integrated models to quantify ES impacts by assessing the interactions between ecological and human systems. In this sense, Arbault et al. (2014) used the GUMBO model (Boumans et al. 2002) which "simulates context-dependent cause-effect chains between the natural sphere and the human sphere" to retrieve time- and scenario-dependent CFs, while Othoniel et al. (2019) "developed an integrated system dynamics model of land cover and ecosystem services" building on the MIMES framework (Boumans et al. 2015) to develop CFs at the scale of Luxembourg and its municipalities. Both studies show the benefits of using system dynamics to quantify CFs to assess impacts on ecosystem capacity to supply services. Nevertheless, the generated CFs are spatial and/or temporal context-dependent and, therefore, their use cannot be easily generalized. Only the approach can be replicated, ideally, on a per-country or even regional basis.

Among these different clusters, the cascade model has been endorsed by several studies to support the development of ES-related CFs (Pavan and Ometto 2018; Maia de Souza et al. 2018; Rugani et al. 2019). Although Rugani et al. (2019) suggested guidelines for the computational structure, first studies that build on the cascade model are emerging (Pavan and Ometto 2018; Liu et al. 2020; Alshehri et al. 2023; Lago-Olveira et al. 2025).

Despite all these advances, the integration of ES into LCIA remains limited. Firstly, the overall coverage of ES impacts in LCIA is low (Othoniel et al. 2016; Alejandre et al. 2019; Rugani et al. 2019; Vanderwilde and Newell 2021; De Luca Peña et al. 2022; Hardaker et al. 2022)). Although ES classifications such as CICES intend to be exhaustive, only a limited number (mainly terrestrial) of ES have been addressed in LCIA (Alejandre et al., 2019; Vanderwilde and Newell, 2021). Cultural ES, in particular, have been overlooked or completely excluded (Alejandre et al. 2019; Vanderwilde and Newell 2021; Soldati et al. 2023). Secondly, the focus has predominantly been on land use as a key pressure on ES, while other critical environmental pressures are underrepresented (Hardaker et al., 2022; Rugani et al., 2022).

Finally, analysis of the various proposals to integrate ES into LCIA models through the lens of ES knowledge reveals a lack of terminological clarity regarding ES concepts. The reviewed LCA literature often refers to ES without specifying whether they consider ecosystem functions, ES flows, or benefits (Othoniel et al., 2016; Pavan and Ometto, 2018; Hardaker et al., 2022). Given the conceptual difference between these concepts, underlined in the cascade model, the absence of a clear terminology hampers positioning the different proposals for ES integration in LCIA, and the understanding of the entity modelers intend to assess.

5.6. Metrics for assessing potential impacts on ES at the damage level are often unclear, conductive to potential inconsistencies.

Authors have suggested assessing ES at the damage level using various metrics such as emergy (Odum 1996), human welfare (Arbault et al., 2014) or monetary valuation (Cao et al. 2015; Bruel et al. 2016; Alejandre et al. 2019; Othoniel et al. 2019). Monetary valuation is often proposed to express the costs of losing ES for society (Bulle et al., 2019). Nevertheless, there is no recommendation on the monetary valuation method to use while "different valuation methods may not be measuring the same economic construct and therefore values from different methods may not be directly comparable" (De Groot et al. 2002). ES-oriented characterization models at the damage level (e.g. Cao et al. (2015); Othoniel et al. (2016); Charpentier Poncelet et al. (2021); Ardente et al. (2023)) frequently employ, and sometimes combine, different monetary valuation methods to account for both damage and adaptation costs. Although economists generally accept aggregating results from multiple methods, ensuring transparency in method selection is crucial for maintaining consistency across models used in comparative analyses.

5.7. How ES interact with existing AoP lacks clarity.

The impacts mechanisms on human health, ecosystem quality, and ES are interrelated and should not be modeled in silos (Schaubroeck and Rugani 2017).

Important research efforts have been conducted to assess the role of biodiversity in the supply of ES, leading to the development of the ecological subfield of biodiversity-ecosystem functioning (BEF) (Balvanera et al. 2006; Cardinale et al. 2012; Tilman et al. 2014; Isbell et al. 2017). BEF researchers have evidenced the influence of biodiversity on ecosystem functioning, showing that loss of species is likely to disrupt patterns of material and energy flows, underpinning ecosystem functioning (Isbell et al. 2017; Jochum et al. 2020; Hong et al. 2022). Callesen (2016)) notes that the loss of some species that play an essential and quantitatively important role in the structure and processes in ecosystems, called *keystones* species, "may trigger a cascade of effects on ecosystem structures and functions", resulting in a loss of ES. As a result, the ability of ecosystems to provide services might be altered. . The link between the ecosystem quality AoP and ES has already been discussed by LCIA scholars (De Souza et al. 2013; Koellner and Geyer 2013; Woods et al. 2018; Oginah et al. 2023). However, outside of the recent approach suggested by Oginah et al. (2023) (see 5.5.2), current LCIA practices and characterization models hardly express this interconnectivity. Clarifying these links in LCIA could shed light on current gaps and offer interesting research avenues for LCIA method developers. Since characterization models pointing to the AoP Ecosystem Quality result from a transformation of ecosystem functions, it is likely that these cause-effect chains can be extended towards the ES AoP.

Ecosystems are also closely related to human health. By definition, ES play a crucial role in supporting human health, directly and indirectly (Costanza et al. 1997; Sandifer and Sutton-Grier 2014). This link is clear for ES such as food provisioning or pest control. However, less direct contributions can be acknowledged ; human contact with biodiversity and nature can result in psychological and physiological health benefits (Sandifer et al. 2015; Fisher et al. 2023; Zhou et al. 2024).

Thus, by influencing the supply of ES, a change in ecosystem functions can affect human health. However, one of the main challenges is to determine how to quantitatively account for the effect of such a change and integrate it into characterization models in a manner that ensures both reproducibility and practical applicability.

Some characterization models already feature links between the AoP. Cao et al. (2015) assess the potential impact of land use on the capacity of the ecosystem to filtrate water as the cost to society to adapt to this loss of service. The authors distinguish countries that are economically able to adapt (having to assume economic costs to avoid losing service) from others. When not able to adapt, countries face potential impacts on Human Health due to a loss of water provisioning. Although this complementary impact pathway has not yet been operationalized, this proposal highlights that impact scores and affected AoP can vary depending on whether adaptive or nonadaptive behaviors are considered. It is therefore essential for LCIA method developers to explicitly state whether adaptation measures are accounted for and to ensure consistency in their modeling assumptions and choices.

6. Conclusion

Despite growing interest in integrating ES into LCIA models and factors, progress remains hindered by conceptual, methodological, and terminological inconsistencies. Although natural resources and provisioning ES are inherently connected, they are often treated separately, resulting in fragmented approaches within LCIA models.

Various proposals have been made to integrate ES as a new AoP or within existing AoP, but the absence of consensus is conducive to the development of inconsistent models which follow distinct modeling objectives. Despite several proposals to incorporate ES into characterization models, these efforts are frequently undertaken following different schools of thought, without harmonized guidelines to unify these approaches. In addition, different valuation methods used to assess the potential impacts on ES at the damage level address varying aspects of ES, but the rationale behind their selection is rarely discussed or explicitly justified. Finally, critical interactions between ES concepts and existing AoP remain insufficiently addressed in current LCIA practices, despite representing promising avenues for further research and development. As a result, the integration of ES in LCIA remains underdeveloped and inconsistent, with limited coverage across different categories of ES and types of environmental pressure.

This critical review attempts to establish a foundational understanding of ES concepts for the progress of LCIA, identifying the most urgent challenges to account for the ES value in impact assessment models. Its conclusions call for the development of a comprehensive, consistent, and globally accepted framework to facilitate the integration of potential impacts on ecosystems' capacity to provide services at the damage level in LCIA. By establishing a structured approach to modeling potential impacts on ES, the LCIA community can better capture the complexity of ES, ultimately supporting more informed decision making.

Data Availability

Data sharing is not applicable to this article.

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